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Comparison of trap and underwater video gears for indexing reef fish presence and abundance in the southeast United States

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ABSTRACT

It is challenging to manage reef fish species in the Southeast U.S. Continental Shelf Large Marine Ecosystem (SUSLME) due to life history strategies that make them vulnerable to overexploitation, difficulty of sampling reef fish in high-relief hard bottom habitats, and fluctuations in utility of fishery-dependent data. In response to declines in fishery-dependent data due to fishery closures, fishery-independent sampling of reef fish has become even more critical to stock assessment. Here we test whether a long-term chevron trapping survey could benefit from the addition of underwater video cameras. Sampling occurred on continental shelf and shelf break habitats (15–83 m deep) between northern Georgia and central Florida. Reef fish frequency of occurrence was significantly higher on video compared to traps for 11 of 15 species analyzed, and the increase ranged from 38% to infinity for these 11 species. Frequency of occurrence for the four remaining species was not significantly different between traps and video. Although positive relationships were observed between log-transformed trap and video indices of abundance for five selected reef fish species, considerable amounts of unexplained variation existed and the relationship for three species was nonlinear. Underwater video can be a beneficial addition to a long-term trapping survey by increasing the frequency of occurrence for most reef fish species, which should translate into improved indices of reef fish abundance in the SUSLME.

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1. Introduction

It is challenging to manage economically important reef fish species in the Southeast U.S. Continental Shelf Large Marine Ecosystem (SUSLME; North Carolina to Florida). The life history characteristics of many reef species – e.g., slow growth, long life span, late maturity, high site fidelity, complex social structure, and the ability to change sex – render them vulnerable to overexploitation (Coleman et al., 1999). Reef fish species can be difficult to sample because they are often found in high-relief habitats and sometimes in strong currents (Powles and Barans, 1980; Grimes et al., 1982). Assessment is complicated by intermittent commercial fishing operations, multiple landing sites, dynamic regulations, and the overall reduction of robust fishery-dependent data due to fishery closures. Reductions in fishery-dependent data have drastically increased the importance of, and need for, spatially- and temporally-extensive fishery-independent data (Jardim and Ribeiro, 2007). Prohibiting possession of a species removes nearly all of the fishery-dependent data sources for the stock, making it virtually impossible to evaluate the stock status post-closure; in this case, decisions on when or if to remove the closure are entirely reliant upon fishery-independent data (Walters and Martell, 2004).

A wide variety of fishery-independent methods have been used to index reef fish abundance in the SUSLME. Some of the methods explored include larval sampling during estuarine ingress (Adamski et al., 2011), estuarine trawling in seagrass beds for juveniles (Koenig and Coleman, 1998; Adamski et al., 2011), standardized hook-and-line sampling (Bacheler and Buckel, 2004; Rudershausen et al., 2008), bottom longlining (Wyanski et al., 2000), fisheries acoustics (Rudershausen et al., 2010), and fish traps (Cuellar et al., 1996; McGovern et al., 1998). Most of the fishery-independent sampling data for reef fishes in the SUSLME has been collected by the Marine Resources Monitoring, Assessment, and Prediction (MARMAP) program run by the South Carolina Department of Natural Resources. In addition to longlining, MARMAP has used chevron traps since the late 1980s to survey reef fish species on

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the continental shelf between North Carolina and Florida. MARMAP chevron trap data have been used to generate indices of abundance for stock assessments for several species (e.g., black sea bass *Centroprisis striata*, red porgy *Pagrus pagrus*, vermilion snapper *Rhomboplites aurorubens*, red grouper *Epinephelus morio*), but chevron trap indices for other high-profile species (e.g., red snapper *Lutjanus campechanus*, gag *Mycteroperca microlepis*) have not been useful because low overall catches resulted in large coefficients of variation around annual relative abundance estimates. It is not uncommon for fish trapping surveys to have insufficient sample sizes and highly variable catches for some species, and in these cases surveys often lack the power to detect significant changes in population abundance over time (Cappo et al., 2003).

In 2010, the National Marine Fisheries Service (NMFS) created the SouthEast Fishery-Independent Survey (SEFIS) to work with MARMAP to increase fishery-independent sample sizes in the SUSLME, as well as evaluate additional sampling gears for indexing reef fish in the region. The first new gear evaluated by SEFIS was underwater video, which has been used to index the abundance of fish species in places such as the Gulf of Mexico (Gledhill et al., 1996; Gledhill, 2001), Hawaii (Ellis and DeMartini, 1995), Alaska (Stoner et al., 2008), Australia (Harvey et al., 2007), and the deep-water abyssal plains (Priede and Merrett, 1996). In the SUSLME, underwater video has not been used to quantify the likelihood of fish being available to traps and also caught (i.e., trap detection probability). In other words, video may help determine if zero catch in a trap is due to fish being (1) truly absent from the area or (2) present around the trap but not caught. However, if reef fish information from traps and videos is redundant, the implementation of underwater video to index reef fish species may not be justified.

Our first objective was to compare the frequency of occurrence and abundance between traps and underwater video for several economically important reef fish species in the SUSLME. Since much of the variability in models indexing abundance (i.e., delta-lognormal models; Dick, 2004) is due to the binomial (i.e., presence–absence) component, whichever gear has the highest frequency of occurrence of reef fish species will likely produce an index with lower variability and thus will be more useful and informative in stock assessments (Maunder and Punt, 2004). Our second objective was to compare the abundance levels of a subset of reef fish species that were frequently caught in traps and seen on videos. We expected a positive, linear relationship if each gear effectively tracked relative abundance.

2. Materials and methods

2.1. Study area

The SUSLME is a broad expanse of sand and mud bottom interspersed with areas of hard substrates ("hard bottom"), with which most of the exploited reef fish species in the region associate. These hard bottom habitats are quite variable, ranging from high-relief (>10 m) rocky ledges to sand or gravel veneer on flat limestone pavement (Schobernd and Sedberry, 2009; Glasgow, 2010). Hard bottom is sometimes called live bottom because its primary composition, limestone rock, often hosts a diverse epifauna that is important food for reef fish. The major oceanographic feature of the SUSLME is the Gulf Stream, which influences outer sections of the continental shelf as it flows northward. Consistently warm Gulf Stream waters along the outer SUSLME shelf allow tropical and subtropical species to inhabit areas at least as far north as North Carolina (Miller and Richards, 1980). For the current study, sampling occurred on continental shelf and shelf break habitats from central Florida to northern Georgia (Fig. 1).

2.2. Sampling approach

Hard bottom locations were selected for sampling in one of two ways. First, we sampled some sites from the historical MARMAP sampling universe. MARMAP has been sampling reef fishes in the SUSLME using various methods since the 1970s, and has accumulated approximately 1800 potential sampling stations on hard bottom habitat between Cape Hatteras, North Carolina, and St. Lucie Inlet, Florida. Each year, MARMAP randomly selects a portion of these sites for reef fish sampling. In 2010, we sampled some of these randomly selected MARMAP stations off Georgia and Florida, as well as some MARMAP sampling stations that were not randomly chosen. Second, we sampled some additional hard bottom sites that were discovered using various sonar gears (e.g., typical fishing vessel echosounder, split-beam acoustics, or multibeam acoustics). Sampling for this study occurred aboard two vessels, the Skidaway Institute of Oceanography's R/V *Savannah* (28 m long) and the NOAA Ship *Nancy Foster* (57 m long).

Chevron fish traps were deployed at each station sampled in the study. These traps were constructed from plastic-coated galvanized 12.5 ga. wire (mesh size = 34 mm × 34 mm), and were shaped like an arrowhead that measured 1.7 m × 1.5 m × 0.6 m, with a total volume of 0.91 m^3 (Fig. 2; Collins, 1990). Each trap was baited with 24 *Brevoortia* spp. Chevron traps soaked for an average of 105 min (range = 71–174); catch-per-unit-effort (hereafter referred to as "trap index of abundance") for each fish species collected in chevron traps was calculated as number caught per hour soak time.

Underwater video cameras were affixed to each chevron trap deployed in the study (Fig. 2). In our study, high-definition Go-Pro Hero® video cameras were contained in underwater housings and attached over the mouth of the trap, facing away from the trap (Fig. 2). Cameras were turned on and set to record before traps were deployed, and were turned off after traps were retrieved. Traps with corresponding videos are hereafter referred to as 'trap-video samples'. Trap-video samples were excluded from our analysis if videos were unreadable for any reason (e.g., too turbid or dark, camera out of focus, files corrupt) or the traps did not fish properly (e.g., traps bouncing due to waves or current, trap mouth was obstructed). Since depths sampled in our study were always less than 85 m deep, light was only a limiting factor for GoPro cameras when turbidity was high.

Relative abundance of fish species from videos was estimated using the 'MeanCount' approach of Conn (2011). The most common approach to estimate relative abundance of fish species has been the 'MaxN' approach (Ellis and DeMartini, 1995), which is calculated as the maximum number of individuals of the target species observed in any single frame in the video. However, Conn (2011) showed that 'MaxN' may not scale linearly with true abundance; instead, he developed and recommended the 'MeanCount' metric, which is calculated by computing the average number of individuals of a target species in a number of video frames in the video sample. Using simulations, Conn (2011) showed that the 'MeanCount' approach scaled linearly with true abundance with little loss of precision. Based on these findings, we employed the 'MeanCount' metric to estimate relative abundance of target fish species in this paper.

Due to logistical and personnel constraints, we did not count every fish species in each video during video analysis. We limited our list to a total of 107 species that are either assessed by the Southeast Data, Assessment, and Review program or classified as highly migratory species. Lionfish (*Pterois* spp.) was also included on our priority species list due to its recent introduction in the SUSLME and expanding population size (Morris et al., 2011). We limited our video reading to a time interval of 20 total minutes, beginning 10 min after the trap landed on the

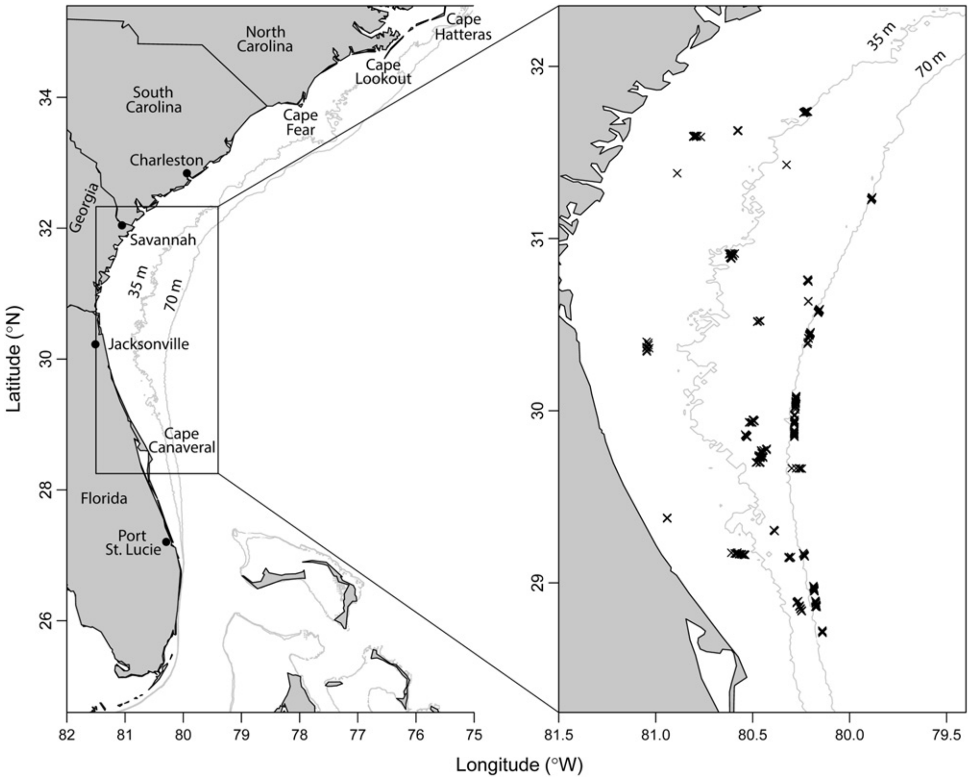

Fig. 1. Study area in the Southeast U.S. Continental Shelf Large Marine Ecosystem where the trap and video survey took place in 2010. Each "×" in the right panel denotes the location of a trap-video sample used in the analysis ($N = 247$); note that symbols may overlap.

bottom to allow time for the trap to settle. We read 1-s snapshots every 30 s for the 20-min time interval (totaling 41 snapshots read). The mean number of individuals of each target species in the 41 snapshots was considered the MeanCount for a video sample; the estimated relative abundance for each species based on MeanCount data is hereafter referred to as the "video index of abundance."

2.3. Data analysis

A common limitation in the development of indices of abundance for fish species is the high proportion of zero catches often encountered (i.e., zero inflation), which can invalidate the assumptions of index of abundance models (Maunder and Punt, 2004). To address zero inflation, we examined the patterns of presence or

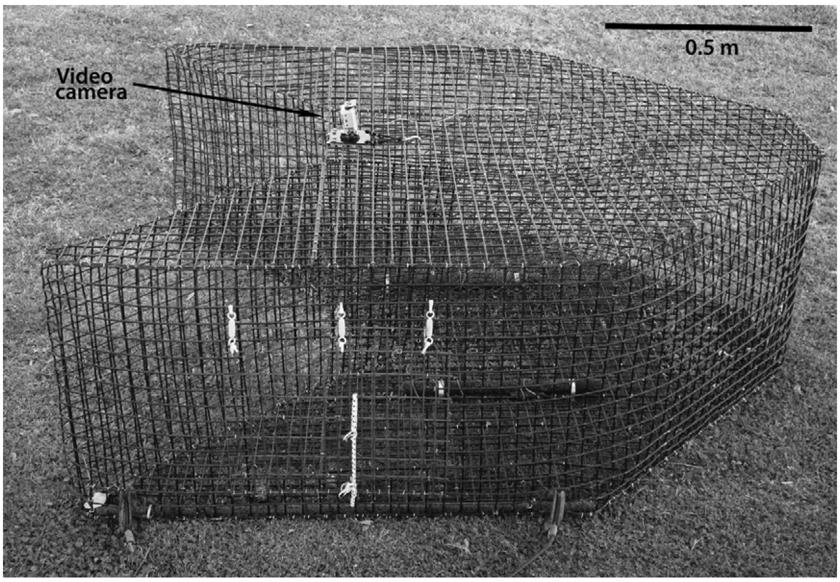

Fig. 2. Chevron trap and attached underwater video camera (high-definition Go-Pro Hero® in an underwater housing) used in the trap-video study in the SUSLME in 2010. Note underwater video camera position over the mouth of the trap, facing away from the trap. Photo credit: Nathan Bacheler.

absence (i.e., frequency of occurrence) for a variety of fish species in the trap and video datasets. We calculated the total number of traps in which each species was caught and the total number of videos in which the species was observed (i.e., MeanCount > 0), and used this information to determine the percent increase or decrease on videos compared to traps as:

$$\%\,\text{increase or decrease on videos} = \frac{v-t}{t} \times 100, \quad (1)$$

where v is the number of videos in which the target species was observed and t is the number of traps in which the species was caught. For each species, we then compared the frequency of occurrence from traps to the frequency of occurrence from videos using an exact binomial test. The two-tailed exact binomial test assesses the null hypothesis that frequencies from two categories (in this case, trap or video) are equal. The exact binomial test was chosen over the Chi-square or *G*-test due to its ability to handle small sample sizes (Sokal and Rohlf, 1995). Only species that were caught in traps or seen on videos more than five total times were included in this analysis; species with five or fewer observations were not helpful in understanding the relative differences between traps and videos because the power to detect differences was so low.

To determine if there were areas where traps or videos did not effectively index fish species, we next examined the spatial patterns in trap and video frequency of occurrence for a subset of species. Five species were examined here, each being present in at least 40 trap and 40 video samples: black sea bass, gray triggerfish *Balistes capriscus*, red porgy, red snapper, and vermilion snapper. To examine spatial differences in species-specific frequency of occurrence patterns, we plotted the frequency of occurrence of each species in the trap and video samples separately.

In addition to addressing the zero inflation issue, we also compared trap (number caught per hour) and video (MeanCount) indices of abundance. Ideally, trap and video indices of abundance would each be compared to true abundance, and linear relationships would indicate that traps and videos index true abundance equally well. However, estimating true abundance in the field is extremely difficult. Instead, we compared trap and video indices of abundance to each other and tested whether these relationships were linear or nonlinear. We fitted a linear model (log-transformed trap index of abundance = $b \times$ log-transformed video index of abundance), a Beverton-Holt model (log-transformed trap index of abundance = [$a \times$ log-transformed video index of abundance]/[b + log-transformed video index of abundance]), and an exponential model (log-transformed trap index of abundance = $a^{\text{log-transformed video index of abundance}}$) to trap-video data for each of the five reef fish species. We used Akaike's information criterion (AIC; Burnham and Anderson, 2002) to determine the most parsimonious model. The AIC method penalizes increases in the number of parameters and rewards goodness of fit, and was calculated as:

$$\text{AIC} = -2\log\ [L(\tilde{\theta})] + 2K, \quad (2)$$

where $L(\widehat{\theta})$ is the likelihood of model θ and K is the number of parameters in the model. For ease of interpretation, we calculated simple differences between each model i and the model with the lowest AIC value (min) as:

$$\text{AIC} = \text{AIC}_i - \text{AIC}_{\min}. \quad (3)$$

The model with the lowest AIC value was considered to be the most parsimonious model and thus the best model in our model set. Last, normalized Akaike weights (w_i) were calculated to better interpret the relative likelihood of each model:

$$w_i = \frac{\exp(-1/2\Delta_i)}{\sum_{r=1}^{R}\exp(-1/2\Delta_r)}, \quad (4)$$

where Δ_i is the ΔAIC value for the ith model and Δ_r is the ΔAIC value for each value in the set of models. Therefore, the w_i is the weight of evidence for model i being the best model in the model set (Burnham and Anderson, 2002).

3. Results

A total of 387 trap-video samples were collected in 2010, but 140 of these were not usable due to a variety of reasons (e.g., trap opening blocked, trap moving, video did not record, video too dark or turbid, video files corrupt). The remaining 247 trap-video samples were distributed throughout the inner and outer continental shelf and shelf break habitats between Savannah, Georgia, and Cape Canaveral, Florida (i.e., 28.81–31.74° N latitude; Fig. 1). Valid trap-video samples were collected during five research cruises between 27 July 2010 and 28 October 2010, in waters 15–83 m deep (Table 1).

Fifteen species with sufficient sample sizes were included in the comparison of frequency of occurrence for trap and video samples (Table 2). The most common species caught in traps were black sea bass (33% of valid trap-video samples), gray triggerfish (29%), vermilion snapper (27%), and red porgy (21%). In contrast, the species most commonly seen on videos were vermilion snapper (43% of valid trap-video samples), gray triggerfish (40%), red porgy (36%) and red snapper (34%). Most species examined (11 out of 15) were seen on videos significantly more often than caught in traps (two-tailed exact binomial tests: $P < 0.05$; Table 2), and their increase on videos ranged from 38% for gray triggerfish to infinity for a number of species that were seen on videos but never collected in traps during the study (e.g., gray snapper *Lutjanus griseus*, hogfish *Lachnolaimus maximus*, lionfish, and nurse shark *Ginglymostoma cirratum*). No significant difference in frequency of occurrence among traps and videos occurred for the remaining four species (Table 2).

There were instances where fish species were caught in traps but not seen on the corresponding videos or, alternatively, seen on videos but not caught in the corresponding traps (Table 3). A total of ten species had at least one instance where they were caught in traps without being seen on the corresponding video, the most instances occurring with black sea bass ($N = 19$) and gray triggerfish ($N = 15$). In contrast, thirteen species had at least one instance where they were seen on videos but not caught in the corresponding trap, with the most instances occurring for red snapper ($N = 50$), vermilion snapper ($N = 46$), gray triggerfish ($N = 42$), and red porgy ($N = 42$; Table 3).

The spatial patterns of frequency of occurrence for some species were different among traps and videos, while for other species they were very similar. The geographic range over which red snapper, and to a lesser extent red porgy and vermilion snapper, were present on videos appeared to be greater than the geographic range caught in traps (Fig. 3). There did not appear to be any systematic geographic bias in the spatial distribution of frequency of occurrence in traps compared to videos for these species, except that traps appeared more likely to miss red snapper in the northern section of our study area (i.e., Georgia; Fig. 3). Black sea bass and gray triggerfish appeared to be caught in traps over the same approximate geographic distribution as seen on videos (Fig. 3).

Positive relationships were observed between log-transformed trap and video indices of abundance for all five species (Fig. 4). However, nonlinear relationships were observed for black sea bass, gray triggerfish, and red porgy. In these three cases, Beverton-Holt models were more parsimonious than linear or exponential models (ΔAIC values > 19 in all cases; Table 4). Alternatively, red snapper and vermilion snapper displayed linear relationships between log-transformed trap and video indices of abundance (Table 4; Fig. 4). Regardless of the type of relationship, a considerable amount of unexplained variation existed for all five species.

Table 1
Cruise information from which trap-video samples were collected off Georgia and Florida in 2010. Latitude and depth ranges refer only to usable trap-video samples.

Vessel	Start date	End date	Days at-sea	Total trap-video samples collected	Usable trap-video samples	Latitude range (° N)	Depth range (m)
Savannah	27-Jul	31-Jul	5	34	28	30.88–31.74	28–49
Savannah	18-Aug	27-Aug	10	125	73	28.83–31.74	19–60
Nancy Foster	14-Sep	25-Sep	12	108	49	30.34–31.63	15–61
Nancy Foster	13-Oct	22-Oct	10	79	61	28.71–29.98	21–61
Savannah	24-Oct	28-Oct	5	41	36	28.88–29.94	35–83
Overall	27-Jul	28-Oct	42	387	247	28.71–31.74	15–83

Table 2
Frequency of occurrence of various fish species either caught in traps or observed on videos in Georgia and Florida waters in 2010. FO = frequency of occurrence; %FO = percent frequency of occurrence; ∞ = infinity. A total of 247 trap-video samples were included in the analysis. Significant differences in the frequency of occurrence for each species in traps and videos were determined using an exact binomial test.

Common name	Scientific name	Trap FO (%FO)	Video FO (%FO)	% increase on videos	*P*
Higher frequency on videos					
Almaco jack	*Seriola rivoliana*	2 (1)	32 (13)	1500	<0.001
Gray snapper	*Lutjanus griseus*	0 (0)	38 (15)	∞	<0.001
Greater amberjack	*Seriola dumerili*	2 (1)	33 (13)	1550	<0.001
Gray triggerfish	*Balistes capriscus*	72 (29)	99 (40)	38	0.046
Hogfish	*Lachnolaimus maximus*	0 (0)	9 (4)	∞	<0.01
Lionfish	*Pterois* spp.	0 (0)	8 (3)	∞	<0.01
Nurse shark	*Ginglymostoma cirratum*	0 (0)	11 (4)	∞	<0.001
Red porgy	*Pagrus pagrus*	52 (21)	88 (36)	69	<0.01
Red snapper	*Lutjanus campechanus*	40 (16)	83 (34)	108	<0.001
Scamp	*Mycteroperca phenax*	1 (<1)	29 (12)	2800	<0.001
Vermilion snapper	*Rhomboplites aurorubens*	67 (27)	107 (43)	60	<0.01
No statistical difference					
Black sea bass	*Centropristis striata*	82 (33)	63 (26)	−23	0.13
Gag grouper	*Mycteroperca microlepis*	3 (1)	3 (1)	0	1.00
Red grouper	*Epinephelus morio*	2 (1)	5 (2)	150	0.45
White grunt	*Haemulon plumierii*	8 (3)	4 (2)	−50	0.39

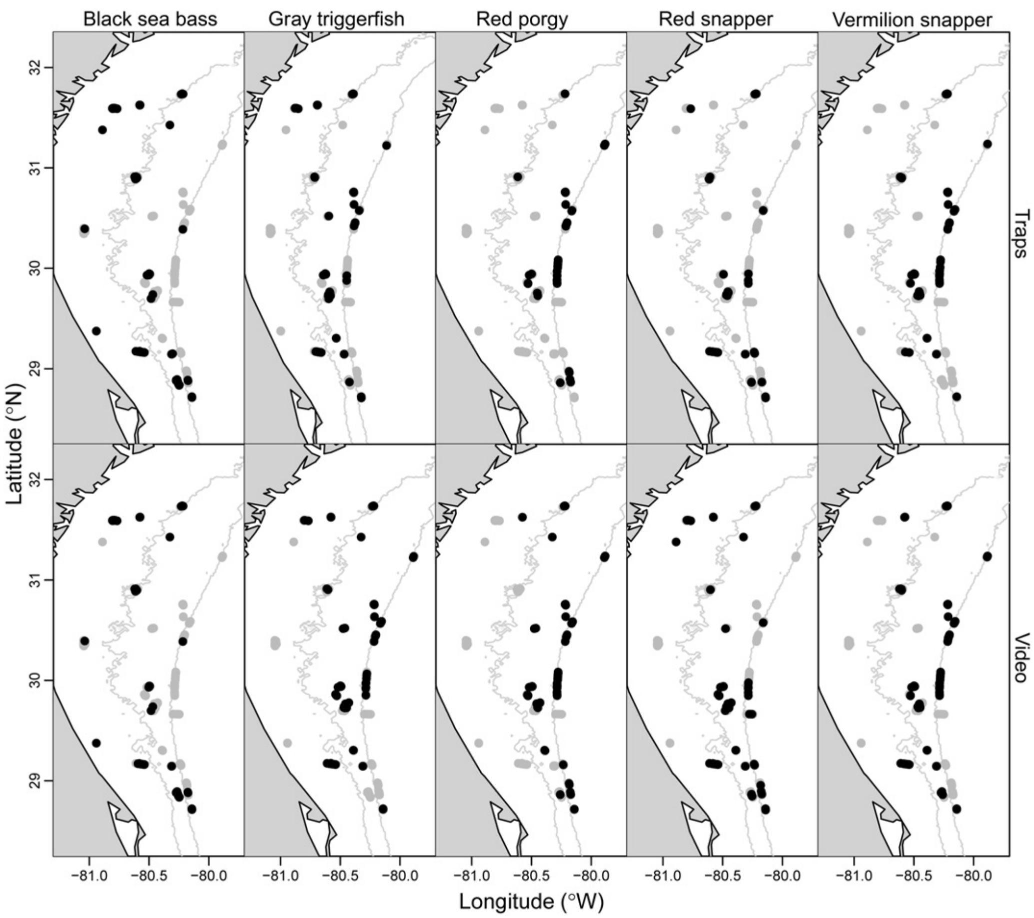

Fig. 3. Spatial distribution of presence (black circles) and absence (gray circles) of five reef-associated species caught in chevron fish traps (top row) or seen on corresponding underwater videos (bottom row) in Georgia and Florida, 2010. Note that many symbols overlap. Bathymetry lines are 35 and 70 m deep.

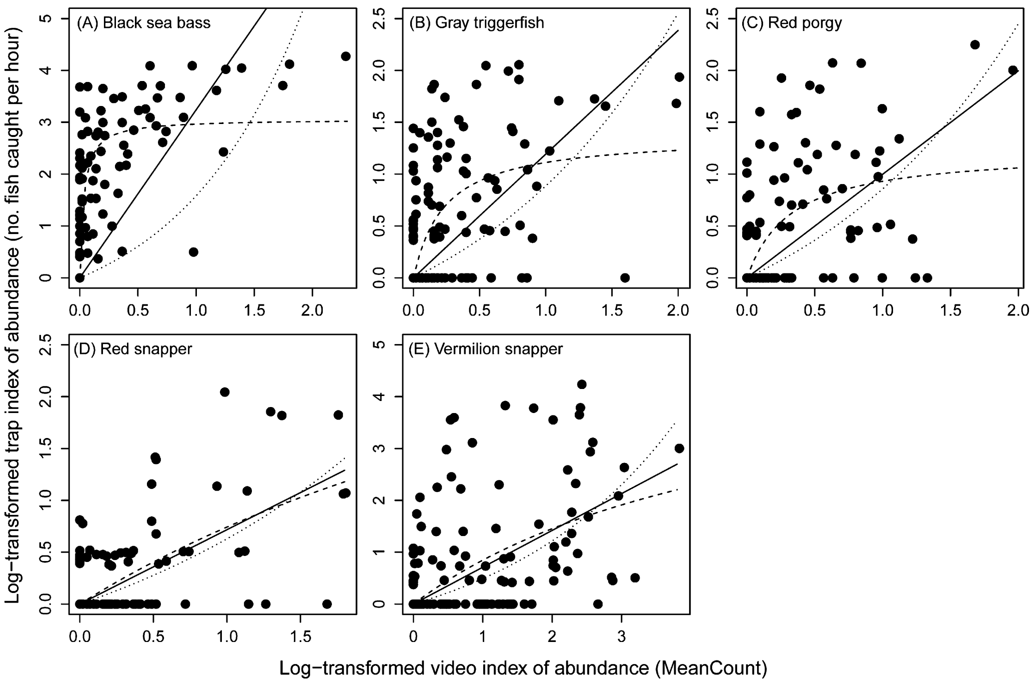

Fig. 4. Log-transformed trap (no. of fish caught per hour) versus log-transformed video index of abundance (MeanCount) for (A) black sea bass, (B) gray triggerfish, (C) red porgy, (D) red snapper, and (E) vermilion snapper in the trap-video survey off Georgia and Florida, 2010. Linear fit indicated by solid line, Beverton-Holt fit indicated by dashed line, and exponential fit indicated by dotted line. Note different axis scales.

4. Discussion

The increasing use of fishery-independent data in fisheries assessment and management has increased attention on determining the most appropriate methods to obtain accurate and precise population data on fish species. The most important element in any sampling design is proper gear selection (Murphy and Jenkins, 2010). Studies comparing multiple gears to index fish species have increased recently as fishery-independent data becomes more important, especially to address questions related to species- and size-selectivity (Willis et al., 2000; Wells et al., 2008; Bacheler et al., 2010). We compared two gears, chevron traps affixed with underwater video, and found that video gear had much higher rates of frequency of occurrence for most reef fish species in the SUSLME. Thus, underwater video will likely be a valuable addition to long-term trapping surveys in the SUSLME and elsewhere.

In multispecies fishery-independent surveys, there is often a high proportion of records in which the catch is zero for many species. Some zeros can be accounted for in delta-lognormal generalized linear modeling approaches (Lo et al., 1992). However, a much larger proportion of zeros often occurs for bycatch or less abundant species, which are often the exact species most in need of a standardized index of abundance (Ortiz and Arocha, 2004). For these species, any survey gear that can reduce the proportion of zero catches in the dataset will likely produce an index with lower variability and thus will be more useful in stock assessments (Maunder and Punt, 2004). In our study, underwater video had a statistically lower proportion of zeros than chevron traps for

Table 3
The number of times each species was caught in traps but not seen on the corresponding video sample, or vice versa, in Georgia and Florida waters, 2010. A total of 247 trap-video samples were included in the analysis.

Species	Present in trap, absent on video	Absent in trap, present on video
Red snapper	7	50
Vermilion snapper	6	46
Gray triggerfish	15	42
Red porgy	6	42
Gray snapper	0	38
Greater amberjack	2	33
Almaco jack	1	31
Scamp	0	28
Nurse shark	0	11
Hogfish	0	9
Lionfish	0	8
Red grouper	2	5
Gag grouper	1	1
Black sea bass	19	0
White grunt	4	0

Table 4
Candidate linear and nonlinear models fitted to log-transformed trap index of abundance versus log-transformed video index of abundance data for five species of reef fish sampled in Georgia and Florida, 2011. Variables are as follows: K = number of estimated parameters; AIC = Akaike information criterion; ΔAIC = AIC differences; w_i = normalized Akaike weights.

Model	Log likelihood	K	AIC	ΔAIC	w_i
Black sea bass					
Beverton-Holt	−226.8	3	459.6	0.0	1.00
Linear	−301.1	2	606.2	146.6	0.00
Exponential	−337.3	2	678.6	218.9	0.00
Gray triggerfish					
Beverton-Holt	−127.9	3	261.8	0.0	1.00
Linear	−147.4	2	298.8	36.9	0.00
Exponential	−161.2	2	326.4	64.6	0.00
Red porgy					
Beverton-Holt	−104.7	3	215.4	0.0	1.00
Linear	−115.5	2	235.1	19.6	0.00
Exponential	−122.5	2	249.0	33.6	0.00
Red snapper					
Linear	−30.1	2	64.3	0.0	0.53
Beverton-Holt	−29.3	3	64.6	0.3	0.46
Exponential	−34.4	2	72.9	8.6	0.01
Vermilion snapper					
Linear	−245.5	2	495.0	0.0	0.57
Beverton-Holt	−244.8	3	495.6	0.6	0.42
Exponential	−250.0	2	504.1	9.1	0.01

11 of 15 species, suggesting that video indices of abundance for these species may have lower variability than trap-based indices of abundance. Similarly, in Western Australia, species richness and abundance of a variety of reef fish species were higher for underwater video compared to fish traps (Harvey et al., 2012).

Higher rates of video frequencies of occurrence for reef fish species would have likely been observed if longer video segments or more snapshots had been read (Gledhill, 2001). Due to various constraints and based on the simulations of Conn (2011), we read 41 snapshots in each video, each spaced 30 s apart, over a time interval of 20 min. Video readers in our study noted some videos where target species such as black sea bass and red snapper were observed in the interval between snapshots or after the 20 min time interval, but not actually present in any of the snapshots included in the MeanCount. Additional resources (i.e., personnel time) for video reading would likely result in even less zero-inflation for reef fish species in the SUSLME, which suggests that the differences in frequency of occurrence we observed between videos and traps are conservative.

There were spatial differences in the likelihood that chevron traps would catch fish when these species were documented on the corresponding video. The most notable example was that red snapper were more likely to be 'missed' by traps in Georgia compared to Florida. It is likely that this geographic difference in trap catchability is related to differences in ambient red snapper densities between Georgia and Florida. In our study, Georgia had a much lower mean video index of abundance (0.49) than Florida (0.79). If the likelihood that traps miss fish is indeed related to overall fish density, our findings support a conclusion that cameras are a more effective gear than traps for documenting the presence of some species (e.g., red snapper) when overall ambient densities are low.

There are several additional benefits of attaching underwater video or still cameras to fish traps in fishery-independent surveys. First, habitat information can be recorded from videos and included in index models so that trap catches or video counts can be standardized to the exact type of habitat sampled (Harris, 1995). Second, videos can be used to identify and subsequently exclude from analysis trap samples that may not fish appropriately, such as traps that bounce or drag in strong currents or that have their mouth opening blocked. Third, additional fish behaviors can be recorded such as spawning, fine-scale habitat use, or feeding behavior. Fourth, evidence is accumulating that behavioral interactions among species in and around traps can strongly influence the catch (Karnofsky and Price, 1989; Jury et al., 2001; Ihde et al., 2006; Ogle and Kret, 2008). For example, Jury et al. (2001) showed that 94% of American lobster *Homarus americanus* entering traps escaped before trap retrieval, and that larger lobsters aggressively defended traps against smaller lobsters, casting doubt on whether lobster catch rates index true abundance accurately. The extent of aggressive interactions in and around the traps by reef fish species is unknown and requires examination. Fifth, underwater videos are nonlethal, which could be particularly useful when indexing rare or rebuilding stocks. Sixth, video may be able to index the abundance of some species that traps rarely catch either due to their large body size (e.g., sharks) or unwillingness to enter traps very often (e.g., lionfish, gray snapper). A final advantage of the video index is that it would be much less influenced by aggressive, social interactions around the trap.

Despite these benefits, it is important to realize that no single gear is able to efficiently collect all the information desired about a fish stock. The collection and reading of underwater video samples, for example, is expensive, time consuming, and requires large amounts of digital memory. Video sampling also cannot provide essential biological data from reef fish species (e.g., ages, spawning period, maturity stage, fecundity, genetics, accurate weight measurements) to stock assessment biologists, information that only an extractive sampling gear like a chevron trap can supply. Additionally, traps can provide samples when videos may be ineffective, such as in excessively dark or turbid conditions. A multi-gear approach to sampling reef fish in the region, such as the one used in this study, takes advantage of the strengths of both extractive and non-extractive methods, providing comprehensive data about reef species in the area while minimizing the effort required to gain a broad spectrum of information.

Stereo-video cameras have been used by various researchers to estimate lengths (and biomass) of fish species, which can then be used to estimate gear selectivity patterns (Harvey et al., 2003; Watson et al., 2005). Stereo-video camera systems are bulky, however, making them difficult to attach to chevron fish traps. Moreover, trap-video sampling gear is occasionally lost in the SUSLME because we often sample in or near Gulf Stream waters with moderate to strong currents, so the high costs of stereo-video systems makes their use too risky in our survey.

The nonlinear relationships observed between trap and video indices of abundance for black sea bass, gray triggerfish, and red porgy are a concern. The goal of any fishery-independent survey is to produce an index of abundance that is linearly related to true abundance (Miller, 1990; Addison and Bell, 1997). We lacked data on true abundance, so we instead related the trap and video indices of abundance for a number of reef fish species, expecting a positive, linear relationship if both index relative abundance well. That the relationships for three species were nonlinear suggests that one or both indices did not track true abundance linearly. One hypothesis that could explain the observed pattern is that these species saturate the chevron trap, exhibiting less rapid catch rates at high ambient densities (Beverton and Holt, 1957). Potential reasons for trap saturation are diverse but could include things such as consumption of all the bait by individuals in the trap, space limitations inside the trap, or aggressive conspecific behaviors by individuals within the trap (Fogarty and Addison, 1997). Conversely, it is also possible that underwater video oversamples these three species at progressively higher abundances, although a mechanism that would lead to such oversampling is not evident. It is likely that underwater video, using the MeanCount approach, is less influenced by saturation effects (Conn, 2011).

We found that underwater video can be a beneficial addition to a long-term trapping survey by reducing zero-inflation for many reef fish species. The improvement of fishery-independent surveys from the addition of underwater video will likely lead to more precise indices of abundance for many SUSLME reef fish species, more robust stock assessments, progress toward ecosystem-based fisheries management, and a better understanding of essential reef fish habitat. Fruitful areas of future research include comparing trap and video indices of abundance to true abundance in well designed field or lab experiment studies, determining the amount of video that needs to be read to accurately and precisely index reef fish, and using video of the trap opening to measure the entry rates, exit rates, and behavioral interactions of reef fish species.

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